Agriculture, Ecosystems and Environment 231 (2016) 320–330

Contents lists available at ScienceDirect

Agriculture, Ecosystems and Environment

journal homepage: www.elsevier.com/locate/agee

Incorporating organic matter alters soil greenhouse gas emissions and increases grain yield in a semi-arid climate

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ARTICLE INFO

Article history:
Received 17 March 2016
Received in revised form 30 June 2016
Accepted 6 July 2016
Available online 6 August 2016

Keywords:
Agriculture
Carbon sequestration
Nitrous oxide
Methane
Semi-arid climate
Soil organic carbon

ABSTRACT

Increasing soil organic matter (OM) is promoted as a strategy for improving the resilience of coarse-textured cropping soils in semi-arid climates. While increasing soil OM can benefit crop productivity, it can also enhance nitrous oxide (N_2O) emissions in temperate climates. Our objective was to investigate if increasing soil OM affected soil greenhouse gas (GHG) fluxes and grain production in a semi-arid region in south-western Australia. We firstly measured N_2O and methane (CH_4) fluxes from a free-draining sandy soil with contrasting soil OM content for 2.5 years using automated soil chambers. The randomized block design included two OM additions (no OM, plus OM) by two nitrogen (N) fertilizer rates (0, 0N; 100 kg N ha^{-1} yr^{-1}, +N) by three replicate plots. Organic matter (chaff) had been applied to the plus OM treatments every three years since 2003, with 80 t OM ha^{-1} applied in total. Secondly, we investigated the interaction between soil OM content and N fertilizer addition on grain yield for two growing seasons. The randomized split-plot design included two OM treatments by five N fertilizer rates (0, 25, 50, 75 and 100 kg N ha^{-1}), by three replicates. Increasing soil OM increased grain yields and soil mineral N, but also enhanced soil N_2O emissions. Nitrous oxide emissions were low by international standards (<0.12% of the N fertilizer applied), with total N_2O emissions after two years ranked: plus OM (+N; 427 g N_2O-N ha^{-1}) > plus OM (0N; 194 g N_2O-N ha^{-1}) > no OM (+N; 41 g N_2O-N ha^{-1}) = no OM (0N; 14 g N_2O-N ha^{-1}). Increasing soil OM also decreased CH_4 uptake by 30%. Management practices that increase soil OM in sandy-textured rainfed, cropping soils in semi-arid regions should be encouraged as they can improve grain yield without substantial increases in soil N_2O or CH_4 emissions.

1. Introduction

Increasing soil organic matter (OM) has multiple benefits for agricultural soils in semi-arid climates. It has the potential to improve the resilience of coarse-textured cropping soils to drying climates by increasing water infiltration, water-holding capacity and nutrient retention capacity (Hoyle et al., 2011), in turn benefiting grain yield (Chen et al., 2013; Lehtinen et al., 2014; Liu et al., 2014). Increasing soil OM is also a strategy for sequestering carbon dioxide (CO_2) to mitigate anthropogenic greenhouse gas (GHG) emissions. Agricultural management practices such as the conversion from conventional to no-till or reduced tillage, residue retention and addition, and crop rotations all have the potential to increase soil OM (Smith et al., 2000; Luo et al., 2010; Liu et al., 2014). It is estimated that widespread adoption of these recommended practices would increase soil OM in croplands by 0.4–0.8 Pg C per year globally (Lal, 2004). However, while these agricultural practices may benefit soil carbon (C) sequestration, their contribution to mitigating global climate warming may be offset by GHG emissions (Six et al., 2004; Liu et al., 2014).

Increasing soil OM can enhance nitrous oxide (N_2O) emissions, a potent GHG, by increasing the availability of nitrogen (N) and C to soil microorganisms (Stehfest and Bouwman, 2006). Crop residues are subject to N mineralization, and in turn nitrification and denitrification; microbial processes that lead to N_2O production (Butterbach-Bahl et al., 2013). For example, nitrifying microbes convert soil ammonium (NH_4^+) to nitrate (NO_3^-) under aerobic conditions, which may result in N_2O as a by-product of the N transformation. Likewise, anaerobic denitrifiers sequentially reduce N oxides (e.g. NO_3^-) to nitric oxide, N_2O and finally N_2; with N_2O emissions resulting from an incomplete conversion. Soil methane (CH_4) uptake could also be inhibited should increasing

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http://dx.doi.org/10.1016/j.agee.2016.07.004

soil OM result in increased soil N concentrations (Le Mer and Roger, 2001). While soil C sequestration would be expected to reach a maximum threshold (Ingram and Fernandes, 2001), the effects on soil GHGs emissions could continue if soil OM concentrations were maintained. Understanding the interaction between increasing soil OM and GHG emissions is critical when assessing the effectiveness of land management practices to abate GHG emissions from the agricultural land sector. However, such studies do not appear to have been conducted in rainfed cropping systems in semi-arid regions.

Increasing soil OM can also increase crop production and has the potential to lower crop fertilizer requirements. International meta-analyses have demonstrated the incorporation of straw and crop residues into soil benefits crop yield with time by improving plant nutrient availability [N, phosphorous (P) and potassium (K)] and soil physical conditions (Chen et al., 2013; Lehtinen et al., 2014; Liu et al., 2014). Realising the benefits of increased soil OM on plant nutrient availability would benefit farmers in semi-arid regions, as fertilizer applications can be a significant expense for grain production. Furthermore, better matching N fertilizer inputs to crop demand not only increases farming profitability, but also decreases the risk of soil N_2O emissions (Millar et al., 2010). Understanding how increasing soil OM influences N fertilizer requirements in semi-arid regions would optimise N fertilizer use, and potentially minimise N_2O emissions, from grain production.

Approximately 17 million ha of semi-arid land in south-western Australia is cropped to cereals, oilseeds and grain legumes on an annual basis, contributing 40% to Australia's annual grain production [Australian Bureau of Agricultural and Resource Economics (ABARE); http://www.agriculture.gov.au/abares; accessed: 16 March 2016]. Mineral N fertilizer is used throughout the region to a varying extent (0–100 kg N ha^{-1} yr^{-1}), with many soils reaching less than 60% of their modelled attainable C storage capacity under a range of cropping systems (Hoyle et al., 2013). The effect of increasing soil OM on GHG emissions and N fertilizer requirements is poorly understood for rainfed cropping soils in semi-arid regions. Instead our knowledge is mainly derived from studies in temperate climates (Stehfest and Bouwman, 2006; Chen et al., 2013; Lehtinen et al., 2014; Liu et al., 2014). Consequently the overall objective of this study was to investigate if increasing soil OM affects soil GHG emissions and grain production in a semi-arid region in south-western Australia. Firstly, we investigated if increasing soil OM altered soil GHG emissions by continuously measuring sub-daily N_2O and CH_4 fluxes from a cropping soil for 2.5 years. Based on the above observations in temperate climates we hypothesized that increasing soil OM would increase soil N_2O emissions and decrease soil CH_4 oxidation. Secondly, we investigated if increasing soil OM altered grain yield and N fertilizer requirements. We hypothesized that increasing soil OM would increase grain yield and lower the amount of N fertilizer required for crop production.

2. Material and methods

2.1. Soil and site

We investigated the effect of increasing soil OM on GHG emissions and grain yield at Buntine (30.01 S, 116.34 E; elevation of 315 m) in south-western Western Australia, 296 km north-east of Perth. The site was established in 2003 using a randomized block design, where each of the three replicate blocks contained a variety of treatments aimed at altering soil OM concentrations; the two treatments of interest to this study were chaff addition (plus OM) and no chaff (no OM) addition. Each replicate treatment plot was large (800 m^2), and divided into sections for use in the present study (see below for details). While applying chaff is not an economic land management practice in the region, it was a means of rapidly increasing soil OM for experimental purposes. Organic matter (chaff) had been applied (20 t OM ha^{-1} per application) to the plus OM treatments every three years since 2003; all treatments were tilled annually each autumn (fall) prior to, and during, the present study. The most recent application of OM was applied and incorporated approximately three months (15 March 2012) prior to commencing the present study, with a total of 80 t ha^{-1} applied to the plus OM treatments since the study site was established. No further OM was applied for the remainder of the study. Consequently the soil organic carbon (SOC) in the surface 100 mm averaged 16 g C kg^{-1} (standard error, 1.2 g C kg^{-1}) in the plus OM treatment versus 10 g C kg^{-1} (standard error, 0.4 g C kg^{-1}) in the no OM treatment during the course of the present study.

The study site was located in a semi-arid region of Western Australia. The nearby town of Dalwallinu has an annual rainfall of 291 mm, a mean daily maximum temperature of 26.3 °C and a mean daily minimum temperature of 11.7 °C (calculated using data collected from 1997 to 2015; Commonwealth Bureau of Meteorology, http://www.bom.gov.au/climate/averages/tables/cw_008297_All.shtml; accessed 18 March 2016). The soil at the experimental site consisted of a free-draining sand (Basic Regolithic Yellow-Orthic Tenosol; Isbell, 2002) and was located on flat to gently undulating land (Table 1).

The land-use history (and grain yield where applicable, averaged across treatments) of the site for the five years prior to commencing this study was: 2007, wheat (*Triticum aestivum*, no yield due to poor growth); 2008, wheat (3.02 t ha^{-1}); 2009, lupin (*Lupinus angustifolius*; no yield); 2010, wheat (2.15 t ha^{-1}); 2011, wheat (3.83 t ha^{-1}). The site was exclusively cropped, and not grazed by farm animals following its establishment in 2003.

2.2. Experimental design and approach

The study included two field-based experiments conducted at the same research site, but using separate areas within each treatment plot for each experiment.

2.2.1. Assessing the effect of increasing soil OM on GHG fluxes

In Experiment 1, the effect of increasing soil OM on GHG fluxes was investigated for 2.5 years (7 June 2012 – 5 December 2014)

Table 1
Selected soil properties (0–100 mm) of plots used to measure the effect of organic matter (OM) addition on greenhouse gas emissions at Buntine, south-western Australia.

Soil property	Plus OM	No OM	$LSD_{0.05}$
Nitrogen (%)[a]	1.05 (0.2)	0.62 (0.03)	0.27
pH (0.01 M $CaCl_2$)[b]	6.20 (0.1)	6.31 (0.1)	0.33
pH (H_2O)[b]	6.84 (0.1)	6.93 (0.1)	0.39
Electrical conductivity ($\mu S\,cm^{-1}$)[c]	216 (25)	136 (6.8)	58
Cation exchange capacity (cmol (+) kg^{-1})[d]	4.19 (0.3)	2.28 (0.1)	0.79
Silt (%)[e]	3.2 (0.3)	2.2 (0.2)	0.86
Clay (%)[e]	6.4 (0.2)	4.9 (0.2)	0.59
Sand (%)[e]	90.4 (0.1)	92.9 (0.2)	0.56
Field capacity (g $H_2O\,cm^{-3}$)[f]	0.39 (1.0)	0.33 (1.7)	0.05

Values are means (and standard errors) of at least six replicates.
[a] Total nitrogen content determined by dry combustion of air-dry, finely ground soils using a CNS analyzer.
[b] Soil pH measured in 0.01 M $CaCl_2$ or water or using a glass electrode and a 1:5 soil to extract ratio.
[c] Electrical conductivity measured in water using a probe and a 1:5 soil to extract ratio.
[d] Cation exchange capacity (CEC) (soil <2 mm) measured using a silver thiourea exchange method (Rayment and Higginson, 1992).
[e] Particle size analysis determined using the method described by McKenzie et al. (2002).
[f] Field capacity determined using ceramic suction plates (McKenzie et al., 2002).

using a randomized block design with three replicates. Two plots (plus OM, no OM; 227 m^2 per plot) were established in adjacent blocks (to ensure the chambers were close enough for the automated system to function) and were each divided into six sub-plots. Half the sub-plots (within a plot) were randomly assigned to receive N fertilizer (100 kg N ha^{-1} yr^{-1}, +N), while the remaining plots received no N fertilizer (0 kg N ha^{-1} yr^{-1}, 0N). This experimental design did not use the entire plot area, and a portion of the remaining plot area was used in the second experiment (see below for details). An area within each sub-plot was designated for measuring gas fluxes, while the remainder of the sub-plot was used for soil sampling. Plots were planted to canola (*Brassica napus* cv CBWA Telfer) on 6 June 2012, barley (*Hordeum vulgare* cv Hindmarsh) on 5 June 2013, and oats (*Avena sativa* cv Brusher) on 6 May 2014. Plots received 40 kg ha^{-1} of triple superphosphate [Ca $(H_2PO_4)_2 \cdot H_2O$] inserted to a soil depth of 30 mm at planting in 2012 and 2013, and no fertilizer at planting in 2014. Half the sub-plots in each OM treatment then received a surface application of urea (100 kg N ha^{-1}) four weeks after seeding (10 July 2012, 3 July 2013, 3 June 2014). The N fertilizer application rate was greater than that typically applied by farmers in the local area (e.g., 30–35 kg ha^{-1}; CSBP Fertilisers, personal communication), but within the range of rates applied in region so as to provide the greatest potential for N_2O emissions. Plants were harvested in late spring each year (29 October 2012, 7 November 2013, 4 November 2014) with the stubble retained in the field.

2.2.2. Assessing the effect of soil OM on grain yield

In Experiment 2, the effect of increasing soil OM on grain yield was investigated for two growing seasons (2013, 2014) using a randomized split-plot design with three replicates. The experimental design included two OM treatments (plus OM, no OM) by five N fertilizer rates (0, 25, 50, 75 and 100 kg N ha^{-1}), by three replicates. Incorporating additional OM into the soil since 2003 had increased a number of soil nutrients in the surface 100 mm of plus OM treatment plots by the start of this study, particularly P and K ($P < 0.001$; Table 2). Each treatment plot was planted to barley (*Hordeum vulgare* cv Hindmarsh) on 5 June 2013, and to oats (*Avena sativa* cv Brusher) on 6 May 2014. In 2013, plots received 40 kg ha^{-1} of triple superphosphate inserted to a soil depth of 30 mm at planting, whereas in 2014 no fertilizer was applied at planting. Each OM treatment plot was further divided into 5 sub-plots (40 m^2) four weeks after seeding (3 July 2013, 3 June 2014), with each sub-plot randomly allocated a different N fertilizer rate (top-dressed as urea). New sub-plots were relocated within each treatment plot in the second year so as to avoid residual effects of unused fertilizer from the first year. Plants were harvested in late spring each year, and at the same time as the experimental plots used to measure gas fluxes. An experimental harvester that recorded grain mass was used to measure grain yield from each experimental plot.

2.3. Measurement of N_2O and CH_4 fluxes

Gas fluxes were measured in each treatment plot using soil chambers connected to a fully automated system that enabled simultaneous determination of N_2O and CH_4 concentrations (Breuer et al., 2000; Kiese et al., 2003). Briefly, the system consisted of a gas chromatograph (Texas Instruments, SRI 8610C) fitted with a ^{63}Ni electron capture detector for N_2O analyses, a flame ionization detector for CH_4 analyses, an automated sampling unit for collecting and distributing gas samples, and 12 chambers (one per treatment plot). Chambers (500 mm × 500 mm; clear Perspex) were placed on metal bases inserted into the ground (100 mm), and fitted with a top that automatically opened and closed. Four bases were located in each treatment plot to enable the chambers to be moved to a new position every week so as to minimize the effect of chambers on soil properties and plant growth. The height of the chambers (not bases) was progressively increased with clear Perspex extensions to accommodate crop growth, with a minimum height of 150 mm and a maximum height of 950 mm. The detection limits were estimated to be 0.6 μg N_2O-N m^{-2} h^{-1} for N_2O and 4 μg CH_4-C m^{-2} h^{-1} for CH_4 at a chamber height of 150 mm, and the dilution via leakage was considered negligible.

The automated gas sampling unit enabled N_2O and CH_4 fluxes to be monitored continuously, providing up to eight (hourly) fluxes per day. A full measurement cycle (3 h) consisted of measuring gas fluxes from four chambers at a time, followed by the next four chambers and then the final four chambers. The system was automatically calibrated by standard calibration gases (e.g., 0.5 μL L^{-1} N_2O, 3.9–4.0 μL L^{-1} CH_4, depending on the calibration cylinder; with a variance +/− 5% of each component) during the measurement cycle. Within a set of four chambers, gas samples were collected sequentially from each chamber immediately following chamber enclosure, with a total of four gas samples per chamber during the enclosure period, plus four standard calibration samples, analyzed for N_2O and CH_4 during a 1 h interval; chamber lids re-opened immediately after the last sample was collected for the set of chambers. Consequently, the chambers remained open for at least 1.43 h between each measurement cycle. The chambers were programmed to open if the air temperature in the chamber exceeded a set value (e.g., 42 °C during growing season, 60 °C at other times) or if it rained (>0.4 mm in five minutes) while the chambers were closed. The air temperature within a single chamber per set of four chambers was monitored using a temperature probe (Pt100, IMKO, Ettlingen Germany).

Table 2
Soil chemical analysis (surface 100 mm) of plots used to investigate the effect of organic matter (OM) addition on grain yield at Buntine, south-western Australia.

Soil property	2013		2014		
	Plus OM	No OM	Plus OM	No OM	$LSD_{0.05}$
Phosphorus (mg P kg^{-1})[a]	62 (6.3)	32 (1.4)	65 (2.8)	35 (4.3)	14
Potassium (mg K kg^{-1})[a]	286 (21)	98 (8.7)	292 (9.1)	98 (16)	48
Sulphur (mg S kg^{-1})[b]	49 (11)	30 (7.8)	34 (5.6)	15 (3.7)	24
pH (0.1 M $CaCl_2$)[c]	6.6 (0.09)	6.5 (0.12)	6.5 (0.2)	6.5 (0.2)	0.4
pH (H_2O)[c]	7.2 (0.09)	7.0 (0.09)	7.0 (0.0)	7.0 (0.2)	0.5
Electrical conductivity (μS cm^{-1})[d]	229 (23)	127 (16)	219 (20)	100 (7.6)	57

Values are means (and standard errors) of three field replicates.
[a] Plant available P and K measured using the Colwell method (Colwell, 1965; Rayment and Lyons, 2011).
[b] Plant available S measured using the Blair/Lefroy extractable sulphur method (Blair et al., 1991; Rayment and Lyons, 2011).
[c] Soil pH measured in 0.01 M $CaCl_2$ or in water using a glass electrode and a 1:5 soil to extract ratio.
[d] Electrical conductivity measured in water using a probe and a 1:5 soil to extract ratio.

2.4. Soil properties with time

In the plots used to measure gas fluxes, soil mineral N (NO_3^- +NH_4^+), dissolved organic C (DOC), and volumetric water content (VWC) were measured to explain seasonal variations in N_2O fluxes. The mineral N of the surface soil (0–100 mm) was measured immediately before planting and then 7 and 14 days following planting or the application of N fertilizer. From then onwards, samples were collected every four weeks. One sample was collected from each plot at each sampling date, with each sample containing four combined subsamples. Soil samples were analyzed for DOC every four weeks.

Nitrate and NH_4^+ were extracted from soil samples by adding 80 mL of 1 *M* potassium chloride to 20 g of sieved (<4 mm) field-moist soil and shaking for 1 h. The filtered solution (Filtech No. 282, Filtech Specialised Products, Adelaide, Australia) was frozen until analysis for NH_4^+ and NO_3^- [plus nitrite (NO_2^-)] colorimetrically using a Berthelot reaction for NH_4^+ and the hydrazinium reduction method for NO_3^- on a Skalar San Plus auto-analyzer (Skalar Inc., Breda, The Netherlands) (Downes, 1978). Dissolved organic C was measured by further filtering the extractant through a polyethersulfone membrane (45 μm pore size; Pall Gelman Laboratory, Michigan, USA; Sarstedt, Nümbrecht, Germany) and analyzing for total oxidizable C (Shimadzu, TOC-5000A). Gravimetric soil water content was determined at the same time that soil samples were collected for mineral N analysis, and after drying sub-samples at 105 °C for at least 24 h.

The soil VWC was measured continuously using moisture probes (MP-406, ICT International, Armidale, Australia) inserted in eight of the 12 plots (i.e., two replicates per treatments) at a depth of 100 mm. Values were collected and stored every 0.5 h using a data logger (SL5 Smart Logger, ICT International, Armidale, Australia). Each probe was calibrated in the field at the end of the study by measuring the probe output (mV) at varying soil water contents, and then immediately collecting a soil sample to determine gravimetric water content and bulk density after drying at 105 °C.

In the plots used to investigate the effect of soil OM on grain yield, soil mineral N was measured to depth prior to seeding. The depth increments were: 0–100 mm, 100–200 mm, 200–400 mm, and 400–600 mm. On each sampling date, samples were collected from three locations per plot, and each sample subsequently analyzed for mineral N as described above. At the same time that samples were collected for mineral N analyzes, bulk density was also determined for each depth increment using a gamma–neutron density meter (Holmes et al., 2012).

2.5. Rainfall, soil and air temperature measurements

Micrometeorological data were collected from 16 June 2012 and for the remainder of the study using an onsite weather station located adjacent to the plots used to measure gas fluxes. Rainfall was measured using a 204 mm diameter automated tipping rain gauge (TB3, Hydrological Services, Warwick Farm, NSW, Australia) with a resolution of 0.2 mm, air temperature was measured at 1.2 m height using a platinum resistance element (PT100 RTD, Vaisala, Finland), while soil temperature was measured at 100 mm depth using a sealed platinum resistance element (PT100 RTD, MTL Engineering Australia, Canning Vale, Western Australia, Australia). A data logger (DataTaker 50, Data Electronics, Scoresby, Victoria, Australia) automatically collected and stored all micrometeorological data. Daily rainfall from 1 January 2012–16 June 2012 was obtained from the nearest rain gauge (Buntine) managed by the Commonwealth Bureau of Meteorology.

2.6. Calculations and statistics

Hourly N_2O (μg N_2O-N m^{-2} h^{-1}) and CH_4 (μg CH_4-C m^{-2} h^{-1}) fluxes were calculated from the slope of the linear increase in N_2O or CH_4 concentration during the chamber lid closure period, and corrected for chamber air temperature, air pressure and the ratio of cover volume to surface area as described by Barton et al. (2008). Flux rates were converted to zero if the regression coefficient (r^2) was <0.80 (de Klein and Harvey, 2012). Daily losses for each plot were calculated by averaging hourly losses for that day. Annual or cumulative fluxes for each plot were calculated by integrating hourly losses over time. Cumulative fluxes were calculated for each year, or part thereof: Year 1, 7 June 2012–6 June 2013; Year 2, 9 June 2013–7 June 2014; Year 3, 8 June 2014–5 December 2014. Cumulative annual N_2O and CH_4 fluxes were also converted to CO_2-equivalents using a global warming potential of 298 for N_2O, and 34 for CH_4 (Myhre et al., 2013).

The emission factor (EF) for the application of N fertilizer to the soil was calculated for each OM treatment using:

$$EF = \frac{[N_2O\text{-N total}(+N)] - [N_2O\text{-N total}(0N)] \times 100}{\text{Total N fertilizer applied}}$$

Where, EF is the emission factor (N_2O-N emitted as % of fertilizer-N applied), N_2O-N total (+N) and N_2O-N total (0N) are the mean annual N_2O emissions (kg N ha^{-1} yr^{-1}) from the +N and 0 N fertilizer plots, respectively, and total N fertilizer applied (kg N ha^{-1} yr^{-1}) is the amount of N fertilizer applied in a particular year. Water-filled pore space (WFPS) was calculated by dividing volumetric water content by total porosity (Linn and Doran, 1984). Total porosity was calculated as [1−(bulk density/particle density)] × 100%, and using a particle density of 2.65 g cm^{-3}.

Two statistical techniques were used to analyze the data: analysis of variance (ANOVA) and linear mixed model (LMM). The effect of OM addition on soil properties, and soil nutrients (combined for both years), measured at the commencement of each study was analyzed as factorial experiments using ANOVA. The linear mixed model was also used to determine whether OM addition or N fertilizer application affected cumulative N_2O and CH_4 fluxes each year, whereas total N_2O and CH_4 fluxes after 2.5 years were analyzed using ANOVA techniques. The effect of OM addition and N fertilizer addition on surface soil properties with time was analyzed using repeated measurements of soil parameters in time, not measured at equal time intervals, using LMM. The best fit for soil NH_4^+ and DOC was achieved using a heterogeneous power model, whereas a general power model (not assuming heterogeneity across time) was the best fit for soil NO_3^-. The effect of OM addition on soil mineral N with depth at seeding, where the measurements were taken at six unequal spaced points, was analyzed using LMM. The latter allowed modelling of the measurements correlation with depth, with a heterogeneous power model being the best fit for the covariance structure. Total mineral N in the soil profile (0–600 mm) was analyzed using ANOVA. The effect of OM addition and N fertilizer on grain yield was analyzed using LMM and reflected the nested structure of the design. All statistical analyses were conducted using Genstat 18th edition (http://www.vsni.co.uk; accessed 18 March 2016).

3. Results

3.1. Soil environmental and climatic conditions

A total of 274 mm of rain fell in 2012 (166 mm during the growing season), 269 mm in 2013 (189 mm during growing season), and 244 mm in 2014 (185 mm during growing season) (Fig. 1a); with daily rainfall ranging from 0 to 20.8 mm during the

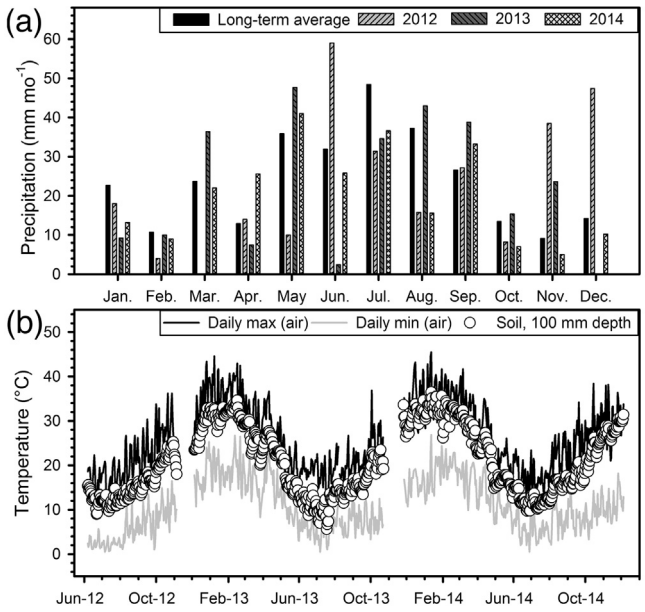

Fig. 1. Monthly precipitation (a), and maximum and minimum hourly air temperature, plus average daily soil temperature (b) with time at Buntine, south-western Australia. Monthly precipitation data was sourced from an onsite weather station and a nearby weather station at the Buntine township.

study (Fig. 2c). Annual rainfall was 85 to 95% of the long-term annual average (285 mm), whereas growing season rainfall was 79 to 91% of the long-term average (206 mm), depending on the year. The distribution of monthly rainfall varied each year (Fig. 1a). For example in 2013, the region experienced a particularly dry winter followed by a 'wet' spring (September–October); whereas in 2014 there was good opening rains in late Autumn (April–May), however this was followed by below average winter rainfall. The mean minimum daily air temperature during the study period was 10.7 °C and the mean maximum daily air temperature was 25.8 °C. Average daily soil temperatures in the surface 100 mm ranged from 5.6 to 36.5 °C, with lowest values recorded in July (mid-winter) and greatest in January (mid-summer) (Fig. 1b).

3.2. Nitrous oxide and CH_4 fluxes

Average hourly N_2O fluxes ranged from −33 to 80 μg N_2O-N m^{-2} h^{-1} during the study and were highly temporally variable (Fig. 2a). Fluxes did not respond directly to the application of N fertilizer, but instead greatest losses occurred in response to summer-autumn rainfall, and to a lesser extent in response to winter rainfall after an extended dry period. Negative N_2O fluxes occurred more regularly with time from the no OM treatments, especially in the absence of N fertilizer.

The total amount of N emitted as N_2O after 2.5 years was greater from the plus OM than the no OM treatments ($P < 0.01$), while applying N fertilizer only increased emissions from the plus OM treatment ($P < 0.05$; Table 3). Consequently, total N_2O emissions after two years were ranked: plus OM (+N; 427 g N_2O-N ha^{-1}) > plus OM (0N; 194 g N_2O-N ha^{-1}) > no OM (+N; 41 g N_2O-N ha^{-1}) = no OM (0N; 14 g N_2O-N ha^{-1}) ($P < 0.05$). Overall treatment effects on annual N_2O losses were similar to those observed for cumulative losses after 2.5 years, except in Year 1 where annual N_2O losses did not vary between the plus OM (0N) treatment and either of the no OM treatments ($P < 0.05$; Table 3). The N_2O fluxes measured following harvest represented 54 to 71% of the annual flux in 2012/2013 depending on the treatment, and at least 97% of the annual flux in 2013/14. The proportion of N fertilizer emitted as N_2O for each one year study period, and after correction for the 'background' emission (0N treatment), varied from 0.08 to 0.12% for the plus OM treatment depending on the year (Table 3). An emission factor for the no OM treatment was not calculated as cumulative N_2O fluxes did not differ between the plus and no N fertilizer treatment for any of the study periods ($P > 0.05$).

Hourly CH_4 fluxes ranged from −211 to 170 μg CH_4-C m^{-2} h^{-1}, with CH_4 uptake dominating over CH_4 emission (Fig. 2b). Methane uptake was observed in all treatments each year, but to a greater extent in the no OM than the plus OM treatments ($P < 0.001$), and was unaffected by the application of N fertilizer ($P > 0.05$). Consequently, total CH_4 uptake after 2.5 years ranged from 724 g CH_4-C ha^{-1} (plus OM) to 1068 g CH_4-C ha^{-1} (no OM) when averaged across N fertilizer treatments (Table 4). Treatment effects on annual CH_4 uptake rates were similar to those observed for cumulative losses after 2.5 years, except for Year 2 where annual CH_4 uptake did not vary between the plus OM (0N) and no OM (+N) treatments (Table 4). Interestingly, CH_4 uptake offset N_2O emissions from the plus OM (0N) and no OM treatments (0N, +N) in the first two years of the study, and as a result these treatments were not a net source of GHG emissions (Table 5).

3.3. Surface soil mineral N, DOC and VWC

The amount of mineral N in the surface soil (0–100 mm) varied seasonally ($P < 0.001$) and in response to either the application of N fertilizer or to summer rainfall (Fig. 3a, b). The amount of NO_3^- in the surface 100 mm ranged from <1 to 62 kg N ha^{-1} and increased in response to both N fertilizer applications and summer rainfall events. Overall, soil NO_3^- tended to be greater in the plus OM than the no OM treatments irrespective of the application of N fertilizer ($P < 0.001$; Fig. 3a). Soil NH_4^+ in the surface 100 mm generally ranged from <1 to 8 kg N ha^{-1}, with the exception of +N fertilizer treatments where soil NH_4^+ contents exceeded 40 kg N ha^{-1} for a brief period following fertilizer applications, returning to baseline values within a month (Fig. 3b).

Dissolved organic C in the surface soil (0–100 mm) varied in response to OM treatment ($P < 0.001$), rather than N fertilizer treatment ($P > 0.05$) (Fig. 3c). Average DOC tended to be greater for the plus OM than the no OM treatments; values ranged from 51 to 227 μg C g^{-1} dry soil for the plus OM treatment (combined N fertilizer rate), and 28 to 158 μg C g^{-1} dry soil for the no OM. Furthermore, DOC peaked during summer and declined during winter with the onset of rainfall.

Soil VWC measured in the surface 100 mm varied on a sub-daily basis in response to rainfall, but in a similar way for all treatments (compare Figs. 2 c and 3 d). Values ranged from <2 to 36% VWC (<5–84% WFPS) during the study, with a median soil VWC of 2.4% (5% WFPS) when averaged across all treatments. Greatest soil VWC values were generally observed during the winter months (June–August), however similar values were occasionally recorded following rainfall in summer.

3.4. Soil mineral N with depth

In the plots used to investigate the effect of soil OM on grain yield, the amount of mineral N in the soil profile (0–600 mm) was always greater in the plus OM than in the no OM treatment prior to seeding ($P < 0.001$; Table 6). In 2013, the difference in mineral N content between the two OM treatments was due to greater mineral N in the surface 100 mm of soil of the plus OM treatment than the no OM treatment ($P < 0.05$); whereas as in 2014, the difference in mineral N content was not attributed to any particular soil depth increment. Interestingly, the amount of mineral N in the soil profile at seeding was at least four times greater in 2013 than in 2014 ($P < 0.05$; Table 6), which was attributed to greater soil N

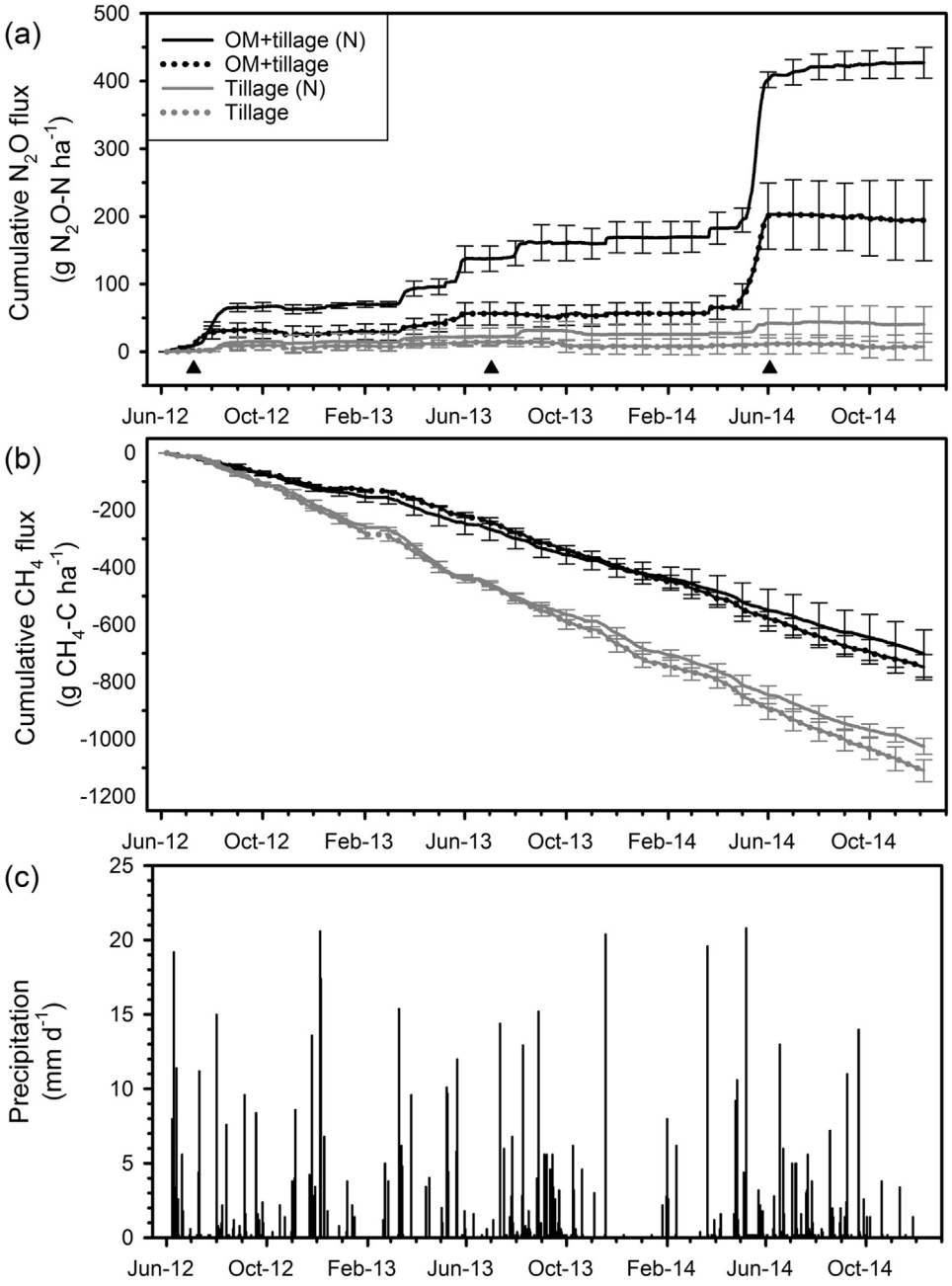

Fig. 2. Cumulative N_2O fluxes (a), cumulative CH_4 fluxes (b), and daily precipitation (c) with time at Buntine, south-western Australia (7 June 2012–5 December 2014). Cumulative N_2O and CH_4 fluxes represent means (±standard errors) of three replicates. The triangles indicate the timing of N fertilizer application. The legend in (a) also applies to (b).

Table 3
Cumulative N_2O flux and emission factor for each organic matter (OM) and nitrogen (N) fertilizer treatment at Buntine, south-western Australia.

Study period[a]	Cumulative N_2O flux (g N_2O-N ha^{-1})[b]				Emission factor (%)	
	Plus OM (+N)	Plus OM (0N)	No OM (+N)	No OM (0N)	Plus OM	No OM
Year 1	138 (19)	56 (17)	21 (12)	14 (6)	0.08	NA
Year 2	271 (36)	146 (47)	19 (18)	0 (10)	0.12	NA
Year 3	18 (6)	−8 (8)	−1 (5)	−7 (7)	ND	ND
All years	427 (23)	194 (59)	41 (26)	7 (19)	ND	ND

Values represent means (and standard errors) of three replicates.
[a] Year 1, 7 June 2012–6 June 2013; Year 2, 9 June 2013–7 June 2014; Year 3, 8 June 2014–5 December 2014; NA, not applicable, as N_2O emission did not differ between the plus and no fertilizer treatment; ND, not determined, as hourly N_2O fluxes not measured for 12 months.
[b] $LSD_{0.05}$ for comparing cumulative N_2O fluxes between treatments in the same year (Years 1, 2 or 3) is 59; $LSD_{0.05}$ for comparing overall cumulative N_2O fluxes (all years) between treatments is 75**;** LSD, least significant difference.

Table 4
Cumulative CH_4 flux for each organic matter (OM) and nitrogen (N) fertilizer treatment at Buntine, south-western Australia.

Study period[a]	Cumulative CH_4 flux (g CH_4-C ha^{-1})[b]			
	Plus OM (+N)	Plus OM (0N)	No OM (+N)	No OM (0N)
Year 1	−251 (39)	−226 (4.7)	−438 (6.0)	−444 (15)
Year 2	−302 (36)	−358 (30)	−411 (25)	−454 (25)
Year 3	−148 (8)	−165 (21)	−176 (4)	−212 (11)
All years	−701 (83)	−748 (44)	−1025 (27)	−1110 (39)

Values represent means (and standard errors) of three replicates.
[a] Year 1, 7 June 2012–6 June 2013; Year 2, 9 June 2013–7 June 2014; Year 3, 8 June 2014–5 December 2014.
[b] $LSD_{0.05}$ for comparing cumulative CH_4 fluxes between treatments in the same year (Years 1, 2 or 3) is 65; $LSD_{0.05}$ for comparing overall cumulative CH_4 fluxes (all years) between treatments is 119. LSD, least significant difference.

Table 5
Nitrous oxide and CH_4 fluxes expressed as CO_2-equivalents for each organic matter (OM) and nitrogen (N) fertilizer treatment after two years[a] of measurements at Buntine, south-western Australia.

OM treatment	N treatment	CO_2-equivalent (kg CO_2 ha^{-1})[b]		
		N_2O	CH_4	Total
Plus OM	+N	73.0 (7.4)	−25.1 (3.4)	47.9 (9.7)
Plus OM	0N	22.4 (9.9)	−26.5 (1.2)	−4.0 (10.8)
No OM	+N	9.9 (7.2)	−38.5 (1.4)	−28.6 (6.0)
No OM	0N	3.2 (5.1)	−40.7 (1.7)	−37.5 (6.8)
$LSD_{0.05}$		24.7	NS[c]	28.0

Values represent means (and standard errors) of three replicates. LSD, least significant difference; NS, not significant.
[a] 7 June 2012–7 June 2014. CO_2-equivalents are calculated using the first two years of data (rather than 2.5 years) due to intra-annual variability in daily N_2O fluxes.
[b] CO_2-equivalents calculated using a global warming potential of 298 for N_2O, and 34 for CH_4 (Myhre et al., 2013).
[c] Multiple comparison of the means is not valid due to the non-significant interaction between OM and N treatments.

mineralization prior to seeding in 2013 than 2014, and in response to summer rainfall.

3.5. Grain yield

In the plots used to investigate the effect of soil OM on grain yield, incorporating OM into the soil since 2003 increased grain yield in both years of study (2013 and 2014) on average by up to 140% ($P < 0.05$) (Fig. 4). Applying N fertilizer did not affect grain yield in 2013, but caused a decline in grain yield in 2014 for both OM treatments ($P < 0.05$). The grain yields reported at the study site each year were consistent with neighboring farms (Liebe Group, personal communication), with particularly low yields reported in 2014 due to below average winter rainfall.

4. Discussion

4.1. Soil OM and GHG fluxes

Nitrous oxide emissions are likely to be enhanced by increasing soil OM in semi-arid soils. In the present study, cumulative soil N_2O emissions from a coarse-textured soil increased by at least 10-fold as a result of increasing soil OM. Our findings from a semi-arid climate are consistent with those reported from other climates. For example, various global meta-data analyses have demonstrated N_2O emissions from upland agricultural soils tend to increase in response to increasing soil OM (Stehfest and Bouwman, 2006) and with the addition of straw (Chen et al., 2013; Lehtinen et al., 2014; Liu et al., 2014). It has been argued that increasing soil OM stimulates the soil biological sources of N_2O, nitrification and denitrification, via increased availability of C and N to these soil microorganisms. This explanation is supported by our observed increase in soil mineral N and DOC in the plus OM treatment in comparison to the no OM treatments. Furthermore a companion study, utilizing ^{14}C and ^{15}N techniques, demonstrated that while incorporating OM at the present study site over time increased organic N retention by up to 2.8 times, it also increased mineral N supply (mineralization) and potential N loss pathways via nitrification to a similar extent (Fisk et al., 2015a). The authors hypothesized that crop residue additions to coarse-textured soils were likely to increase the risk of N_2O emissions by up-regulating the entire soil N cycle (Fisk et al., 2015a), which has subsequently been observed in the present study.

However, soil N_2O emissions from coarse-textured agricultural soils in semi-arid regions can still be expected to be low relative to temperate climates even when land management practices increase soil OM. The greatest soil N_2O losses in response to crop residue incorporation has tended to occur in heavier textured soils containing >10% clay (Chen et al., 2013). Consequently the range of annual N_2O emissions (0–0.27 kg N_2O-N ha^{-1} yr^{-1}) in the present study, with a low clay content (<6.5%; Table 1), was conservative in comparison to values reported for other cropped sites in Australia and overseas. Globally, and across a variety of climatic regions, annual N_2O losses from cropped mineral soils have ranged from 0.3 to 16.8 kg N_2O-N ha^{-1} yr^{-1} (Stehfest and Bouwman, 2006). The annual N_2O emission reported in the present study was also within the range of values (0.04–0.13 kg N_2O-N ha^{-1} yr^{-1}) previously reported for other cropped soils in the Western Australian grain belt (Barton et al., 2008, 2010, 2011, 2013b; Li et al., 2012). Furthermore, the risk of enhancing soil N_2O emissions in the present study region is low due to the limited capacity to further increase soil OM contents beyond current values in surface soil to 100 mm (Hoyle et al., 2013).

The greatest risk of N loss via N_2O emissions in our study region occurred in response to summer and autumn rainfall. This is consistent with previous observations for semi-arid soils, where a large proportion of an annual N_2O emissions occurred between crop growing seasons, when the soil was fallow, and in response to soil wetting following rainfall (Barton et al., 2008; Galbally et al., 2008). Elevated N_2O emissions following summer and autumn rainfall have been attributed to the rapid release of readily decomposable OM to viable microorganisms following wetting of dry soil (van Gestel et al., 1993). These substrates can be derived from non-living OM already present in the soil, and from the death of microorganisms due to rapid changes in water potential (Kieft et al., 1987; Groffman and Tiedje, 1988; Schimel et al., 2007). Soil wetting also increases the diffusion of substrates and the mobility of microorganisms utilizing those substrates (Borken and Matzner, 2009), and can decrease soil aeration (via increased soil water content) resulting in low soil O_2 (due to rapid microbial respiration at high soil temperatures on wetting) sufficient to promote denitrification (Barton et al., 2013a). Consequently in the present study, increased soil water content and DOC coincided with elevated N_2O emissions following summer and autumn rainfall. Interestingly, and in the absence of crop residue and N fertilizer additions, our semi-arid soil progressively became a sink for N_2O emissions, which we expect was a result of the decline in soil mineral N. Others have also reported net N_2O uptake from soils containing low amounts of mineral N (e.g., Clayton et al., 1997; Rosenkranz et al., 2006), and attributed it to denitrifying bacteria using atmospheric N_2O as an alternative electron acceptor to NO_3^- (Rosenkranz et al., 2006).

Increasing SOC through OM additions has the potential to decrease CH_4 uptake from sandy-textured soils in a semi-arid climates. Both CH_4 production and CH_4 uptake (oxidation) can occur in soil, and so CH_4 fluxes from soils are a product of the

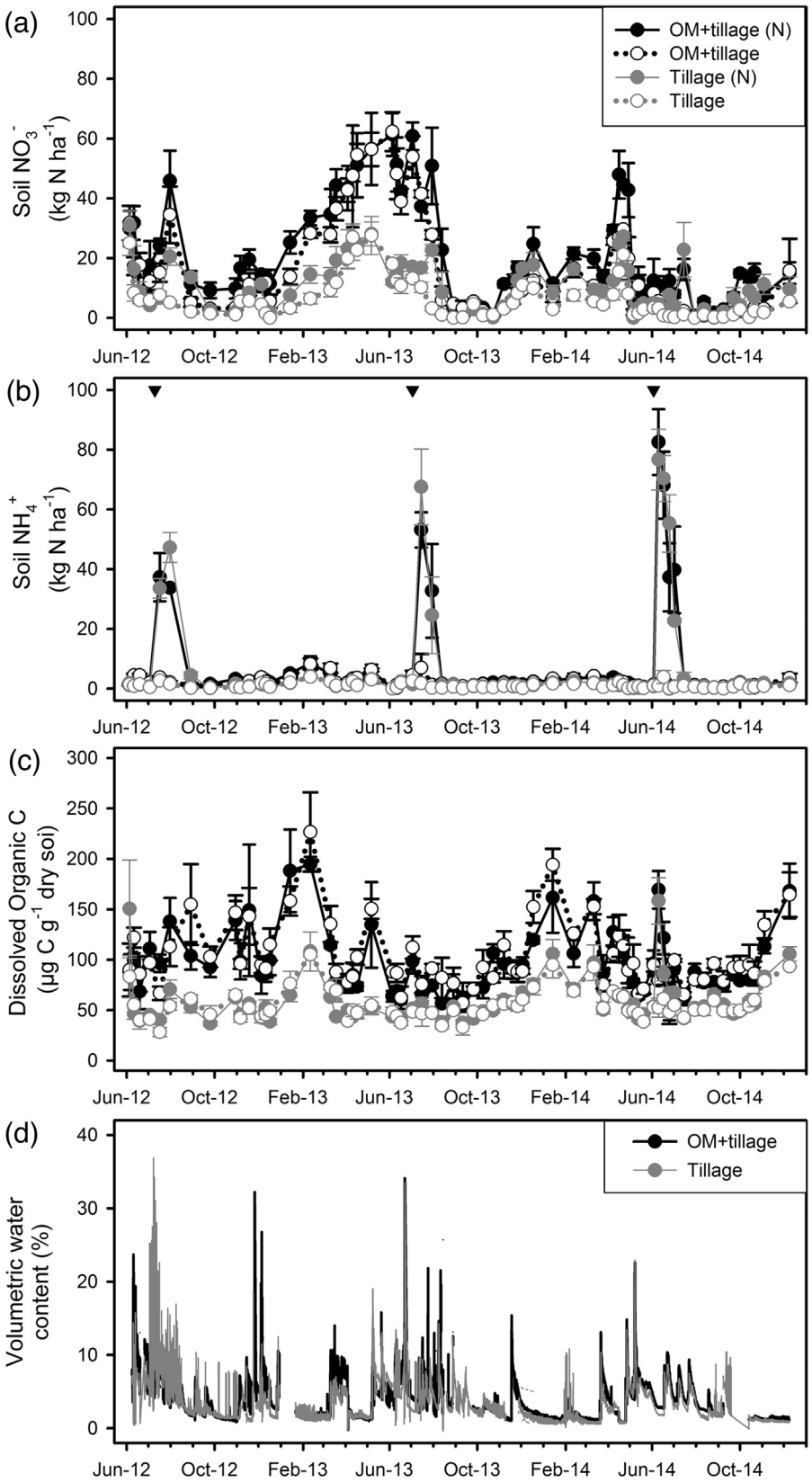

Fig. 3. Soil nitrate (NO_3^-) concentration (a), ammonium (NH_4^+) concentration (b), dissolved organic C (DOC; c), and volumetric water content (VWC; d) in the surface 100 mm of soil with time (7 June 2012–9 December 2014) for each soil organic matter (OM) and nitrogen (N) fertilizer treatment in the GHG emission study at Buntine, south-western Australia. Volumetric water content data for OM treatment averaged across N fertilizer rate. Values represent means (±standard errors) of three replicates, except for VWC data where means are of up to four replicates (errors bar excluded to improve clarity). The triangles indicate the timing of N fertilizer application. The legend in (a) also applies to (b) and (c).

Table 6
Soil mineral N contents (kg N ha^{-1}) with soil depth prior to seeding[a] in plots used to investigate the effect of organic matter (OM) on grain yield at Buntine, south-western Australia.

	2013		2014	
Soil depth (mm)	Plus OM	No OM	Plus OM	No OM
0–100	41 (0.2)	17 (2.9)	9.4 (0.8)	3.8 (0.6)
100–200	31 (3.4)	26 (5.3)	7.4 (1.8)	4.8 (2.2)
200–400	13 (1.0)	10 (0.6)	2.6 (0.1)	2.7 (0.6)
400–600	4.6 (0.8)	2.4 (0.7)	1.2 (0.2)	1.8 (0.2)
Total	90 (3.1)	56 (5.8)	21 (1.6)	13 (2.1)

Values represent means (and standard errors) of nine field replicates.
[a] Soil samples collected on the 6 June 2013 and 7 May 2014.

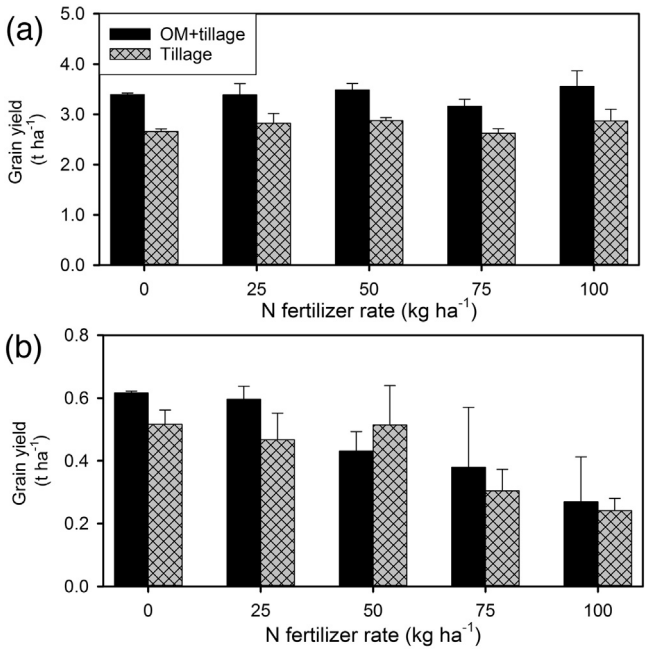

Fig. 4. Grain yield response to varying nitrogen (N) fertilizer application rates for each soil organic matter (OM) treatments in 2013 when sown to barley (a), and in 2014 when sown to oats (b). Data collected from plots used to investigate the effect of soil OM on grain yield at Buntine, south-western Australia. Yields represent means (±standard errors) of three replicates in 2013 and 2 replicates in 2014. The legend in (a) also applies to (b). Note differences in y-axis scale between (a) and (b).

balance (Le Mer and Roger, 2001; Dalal et al., 2008). We hypothesized that incorporation of OM would increase CH_4 production relative to uptake at the present study site, as the resultant increase in soil mineral N in turn would inhibit CH_4 oxidation (Le Mer and Roger, 2001). Alternatively, the increase in soil OM content may have decreased the rate at which air diffused through the surface soil, thus limiting CH_4 oxidation; but given the coarse texture of the soil this is not considered likely. In the absence of OM additions, CH_4 uptake actually offset N_2O emissions in the present study. Methane uptake was also reported from another sandy cropped soil in the same region, but it did not necessarily offset N_2O emissions (Barton et al., 2013b, 2014). The effect of adding crop residues to CH_4 emissions from upland cropping soils has not been studied extensively, with more attention given to 'paddy' soils (Liu et al., 2014). Further research is required to quantify CH_4 uptake in semi-arid regions throughout the world as the present study indicates that there are situations whereby CH_4 uptake may offset other agricultural GHG emissions.

4.2. Nitrous oxide emission factor for rainfed agriculture in semi-arid regions

Increasing soil OM in sandy-textured soils is unlikely to change the emission factor for calculating N_2O emissions from N-fertilized soils in semi-arid environments similar to the present study region. Although increasing soil OM in the present study increased the N_2O emission factor in comparison to values previously reported for the region (0.02–0.06%; Barton et al., 2008, 2010; Li et al., 2012) it was still relatively conservative. The emission factor for the plus OM treatment (0.1%) was 10-fold less than the international default value (1.0%; IPCC, 2006), and half the emission factor used by the Australian Government for calculating N_2O emission from the application of N fertilizer to low rainfall regions (0.2%; Department of Environment, 2013). Furthermore, the emission factor reported in the present study may have been further enhanced by applying N fertilizer above the local average (for experimental purposes) as emission factors tend to increase with an increasing rate of N fertilizer (Millar et al., 2010). Our current, and past, research supports reviewing the emission factors used to calculate N_2O emissions from the application of N fertilizer to non-irrigated, cropped soils in semi-arid regions.

4.3. Soil OM, grain yield and soil mineral N

Increasing soil OM in the surface soil via crop residue additions is likely to benefit grain yield in semi-arid regions, even in years where rainfall is below average. The positive yield response observed in the present study was consistent with international meta-analyses demonstrating that incorporation of straw and crop residues into soil can benefit crop yield over time, as long it does not lower N availability by increasing soil N immobilization (Lehtinen et al., 2014; Liu et al., 2014).

The mechanisms by which increasing soil OM benefits grain yield are not necessarily straight forward. Others have attributed increased crop yield in response to increased soil OM to better soil nutrient availability (in particular N, P and K) and improved soil physical conditions, such as water-holding capacity (Liu et al., 2014). Although the incorporation of additional crop residues increased nutrient availability at the present study site, soil P and K were already sufficient in the non-amended soil (no OM). There was some indication that the increased soil mineral N observed in the plus OM treatments may have benefited grain yield. For example in 2013, applying 25 kg N ha^{-1} to the no OM treatment increased indicators of yield potential at anthesis (i.e., grain head number and dry matter production) to similar values measured in the plus OM treatment that did not receive N fertilizer (data not shown), suggesting grain yield potential in the plus OM treatment benefited from additional N supplied by soil N mineralization. However, the subsequent grain yield showed no response to N fertilizer additions for either OM treatment (Fig. 4), which was also true for both yield potential and yield measurements the subsequent year. Increasing soil OM could have also benefited grain yield by increasing water availability via increased soil water holding capacity, as demonstrated by small differences in soil water content at field capacity between the OM treatments in the present study (Table 1). Irrespective of the mechanism, we expect increasing soil OM is likely to benefit grain yield from sandy, rain-fed cropping soils in semi-arid regions.

4.4. Implications for managing N_2O emissions and N fertilizer inputs in semi-arid environments

Land management practices that increase soil OM in semi-arid cropping soils should be encouraged. Although increasing soil OM increases the risk of N_2O emissions from cropped soils, these losses

are likely to be relatively small from sandy-textured soils. Opportunities for decreasing the risk of N_2O loss following summer and autumn rainfall events in semi-arid regions with summer fallows are limited. Developing strategies for mitigating N_2O emissions from semi-arid cropping regions is more challenging than for temperate climates, as losses occur post-harvest, when there is no active plant growth, and not in direct response to N fertilizer additions (e.g., Barton et al., 2008; Galbally et al., 2008). Strategies that control soil N supply from nitrification, or immobilize excess mineral N via soil microbial or plant uptake, would be expected to decrease the availability of N for subsequent N_2O loss in semi-arid regions. Indeed, previous studies have demonstrated increasing the efficiency of the nitrification process by increasing soil pH (via liming) could in turn decrease N_2O losses from sandy, acidic soils following summer rain (Barton et al., 2013a,b). Our laboratory-based study also suggested nitrapryin, a nitrification inhibitor, may also have the potential to limit N_2O losses in the study region (Fisk et al., 2015b), but requires field-based verification. Controlling the supply and immobilization of mineral N through the incorporation of crop residues has proven to be difficult in the Western Australian grain belt (Hoyle and Murphy, 2011; Fisk et al., 2015a). Instead, the presence of root exudates has been shown to be more effective at increasing microbial N immobilization relative to N supply (nitrification) than long term additions of crop residues (Fisk et al., 2015a). The potential for plants to both utilize mineral N and stimulate soil N immobilization during the summer-autumn fallow, and response to rainfall during these months, therefore warrants further attention in semi-arid, rainfed cropping systems.

Farmers and agronomists should modify N fertilizer inputs to reflect changes in plant available N if increasing soil OM contents increases plant N availability. This may be particularly beneficial in regions such as this study, which experience significant mineralization of soil OM in the summer-autumn fallow and in response to rainfall. In one year, we observed in excess of 90 kg N ha^{-1} in the surface 600 mm of the soil at planting; sufficient to produce 2 t ha^{-1} of wheat without applying additional N fertilizer. However, the extent of soil mineralization prior to planting would be expected to vary from year-to-year in response to soil water content and temperature (Murphy et al., 1998; Hoyle and Murphy, 2011; Fisk et al., 2015a). For example in 2013 at the present study site, and after 91 mm of rainfall over summer and autumn, there was 96 kg N ha^{-1} (surface 600 mm) in the plus OM treatment and 44 kg N ha^{-1} in the no OM treatment at planting (Table 6); however the following year, and after a drier summer-autumn (65 mm), plant available N in both OM treatments at planting almost halved in comparison to 2013. We argue that accurately predicting the supply of mineral N from soil mineralization is critical if the benefit of increased soil C storage for the purpose of mitigating global warming is to be fully realized.

5. Conclusions

Incorporating crop residues to increase soil OM enhanced N_2O emissions, decreased CH_4 uptake, and improved grain yield from a sandy, cropped soil in a semi-arid climate. However, soil N_2O emissions were still low relative to international standards and represented only 0.08–0.12% of the N fertilizer applied. Farmers in the region should be encouraged to employ land management practices that increase soil OM in cropped soils as it increased grain yield in the present study and showed the potential to lower N and other fertilizer inputs. Better predicting the N supply from mineralization will ensure potential fertilizer savings are realized, and adverse losses such as soil N_2O emissions and N leaching are minimized. We believe this is critical if the benefit of increased soil C storage for the purpose of mitigating global warming is to be fully realized.

Acknowledgements

The authors thank D. Donovan for maintaining the automated gas chambers and field site, and C. Swain for coordinating soil and plant analyses. The Liebe Group are thanked for providing access to the Long Term Research Site at Buntine, and assisting with the establishment and maintenance of the study site. Casual assistance was provided by R. Bowles, Y. Sawada and S. Wells and other members of the Soil Biology and Molecular Ecology Group. Angela Stuart-Street completed the soil classification. We thank the two anonymous reviewers for their constructive comments on the manuscript. This project was supported by funding from the Australian Government Department of Agriculture and the Grains Research and Development Corporation. In-kind support was provided by the Liebe Group, Department of Agriculture and Food WA, and The University of Western Australia. D.V. Murphy was supported by the Australian Research Council Future Fellowship Scheme (FT110100246).

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